Harmonizing the assessment of biodiversity effects from land and water use within LCA

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Abstract. Addressing biodiversity impacts in life cycle assessment (LCA) has recently been significantly improved. Advances include the consideration of several taxa, consideration of vulnerability of species and ecosystems, global coverage and spatial differentiation. To allow a comparison of biodiversity impacts of different stressors (e.g. land and water use), consistent approaches for assessing and aggregating biodiversity impacts across taxa are needed. We propose four different options for aggregating impacts across taxa and stressors: equal weight for species, equal weight for taxa and two options with special consideration of species' vulnerability. We apply the aggregation options to a case study of coffee, tea and sugarcane production in Kenya for the production of 1 kg of crop. The ranking between stressors (land vs. water use) within each crop and also of the overall impact between crops (coffee > sugarcane > tea) remained the same when applying the different aggregation options. Inclusion of the vulnerability of species had significant influence on the magnitude of results, and potentially also on the spatial distribution of impacts, and should be considered.

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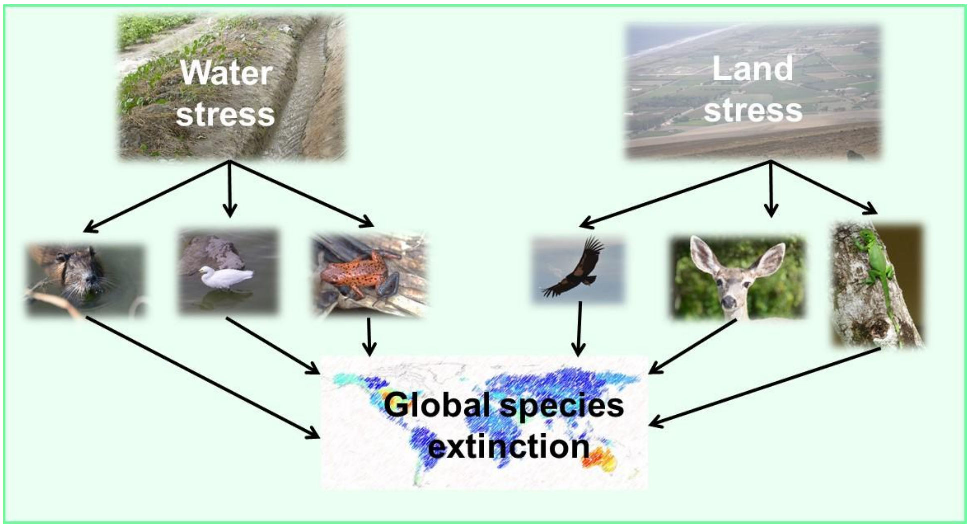


Introduction

Currently the most important driver for biodiversity loss in terrestrial ecosystems is the anthropogenically driven change in land cover.[1] In most cases, this is the conversion of natural systems to cropland, with cultivated systems covering 25% of the terrestrial surface in 2005.[2] Simultaneously, the demand for water has doubled since 1960,[2] and 70% of global water withdrawals are effectuated by agriculture.[3] Half of all inland freshwater ecosystems have been modified or converted during the 20th century.[2] The Global Biodiversity Outlook also states that in the near future a worsening situation regarding extinction risks due to habitat loss is expected.[4]

Life cycle assessment (LCA) assesses potential environmental impacts of products or processes throughout their whole life cycle (i.e. cradle-to-grave).[5-7] Since most product systems nowadays encompass supply chains from various parts of the world,[8] global impact assessment methods are required. The assessment of biodiversity impacts in LCA has for a long time suffered from poor geographic coverage and spatial differentiation.[9] However, recently there has been an important development towards global and spatially differentiated impact assessment methods (e.g. refs[10-13]). These methods include multiple taxonomic groups (e.g. refs[11-14]) and indicators for the vulnerability and irreplaceability of specific species (e.g. ref[12]).

In spite of all these developments, there is no harmonized way of assessing the combined biodiversity impacts of different stressors (e.g. land and water use) in order to better understand environmental trade-offs. An example for such trade-offs is the introduction of irrigation to maximize yields. Irrigation clearly increases biodiversity impacts via water consumption, but at the same time reduces the land areas required for cultivation of the same amount of product. Options for reducing biodiversity impacts could thus consist of a shift of agricultural production to either humid areas (no irrigation required, probably increased impacts of land use on biodiversity) or to more arid areas (probably limited land impacts, but higher water impacts). However, both these options face major challenges in reality regarding the assessment of impacts, due to the difficulty in properly assessing trade-offs between land and water use related impacts and a lack of appropriate inventory data.

The aim of this paper was to suggest ways of harmonizing the assessment of biodiversity impacts from water consumption and land use. We chose the water impact assessment methods from Verones et al.[11, 12] and the land use impact assessment method from de Baan et al.[14] as a basis for this comparison, because both work with similar models (i.e. species-area relationship models) and taxa (mammals, birds, reptiles and amphibians, Table 1). We suggest a framework on how water and land use impacts on biodiversity can be assessed in a harmonized way and propose three options. Furthermore, we analyze how biodiversity impacts of land and water use on different taxonomic groups could be weighted and aggregated. Finally, we evaluate the framework and weighting options in three case studies (coffee, tea and sugar cane production in Kenya).

Materials and Methods

Framework. In LCA, biodiversity impacts are quantified by determining the amount of a pressure (e.g., area of land or volume of water, called inventory flows) required for the production of a product and multiplying this with a characterization factor (CF), which indicates the biodiversity loss associated with the pressure. For land use, two types of impacts are typically distinguished.[15, 16] Land occupation impact quantifies the reduction of biodiversity during the land use phase, i.e. impacts that occur due to continuous suppression of biodiversity recovery while land is occupied by human activities. Transformation impacts consider the time required for an ecosystem to recover after a hypothetical future land abandonment (i.e. transforming natural habitat in slowly recovering ecosystems is considered more detrimental than in fast recovering ones). For water use, currently only the effects of the actual water use (occupation) are considered. For comparability of results, we focus on the harmonization of occupation impacts and exclude land transformation in the comparison of water use and land use impacts in the case study. The assessment of impacts from land transformation in the case study is described in the Supporting Information (SI) S9.

A CF typically consists of a fate factor (FF) and an effect factor (EF). In the method of Verones et al.[11] the FF for water use indicates the lost area of a wetland because of water consumption. The inventory of land use (occupation) is already given as the area used over a certain time and hence the FF is set to 1. The EF indicates the ecological impact (from land and water use) and was separately assessed for different taxonomic groups *t* (mammals, birds, reptiles and amphibians, definition of taxa, see SI S5). The CF is always specific per spatial unit *a*, which, in this paper, is ecoregions[17] for land use and upstream watersheds for water consumption (Table 1). $CF_{total,j,a}$ is calculated for each watershed or ecoregion *a* on a stressor level *j*, i.e. for different land use types and water consumption separately. In order to aggregate the impacts across different taxonomic groups, and to obtain one single CF for biodiversity loss, a weighting factor *W* is introduced (Equation 1).

$$CF_{total,j,a} = FF_{j,a} \cdot \sum_{t=1}^{4}(EF_{j,t,a} \cdot W_t)$$

Equation 1

The EF before aggregation gives impacts in potential global equivalents of species extinction (species-eq) per unit of water consumed (m^3) or per unit of land occupied ($m^2 \cdot yr$).[12] W is applied on an individual stressor basis because of different units (per m^3 or $m^2 \cdot yr$) and transforms the EF from global equivalents of species extinction per taxonomic group to a potentially disappeared fraction (PDF) of species across all taxonomic groups included. The concept of PDFs has been widely used in LCA before (e.g. ref[18]). The fraction of species that was lost is usually calculated based on local and regional losses only and could thus not indicate global species loss (extinction). By contrast, our approach intends to account for *global* losses and thus indicates potentially lost fractions of species on the global scale. In order to avoid confusion we named the PDF for global losses PDF_{global}.

The product of inventory flow ($Inventory_{j,a}$) and CF describes the impact due to stressor j in region a. Because the impact of different stressors is assessed in the same units (PDF_{global}), the total impact due to land and water can now be calculated by adding them up to a total impact score (IS_{total}, Equation 2). The spatial units of inventory flows and CFs may not match. In case the land/water use takes place across multiple ecoregions or watersheds, an area weighted approach is used (Equation 2). This area-weighted approach uses the area shares A_a [-] for that purpose (e.g. A_a is the fraction of the cropland within an ecoregion or watershed a such that $\sum_{a=1}^{m} A_a = 1$).

$$IS_{total} = \sum_{j=1}^{n}\sum_{a=1}^{m} Inventory_{j,a} \cdot CF_{total,j,a} \cdot A_a$$

Equation 2

The calculation of the CF for water use follows a bottom-up approach and involves two steps. Firstly, for the fate factor (FF), the change in area per unit of water removal for more than

20'000 waterbodies is derived via simplified hydrological balances.[11] The reference situation is the present distribution of waterbodies, and impacts are prospectively quantified for a small amount of additional withdrawals in terms of m^2 of wetland area lost per m^3/yr of water consumed. This is also called a marginal approach. It is then followed by the effect factor calculation (species lost per unit of wetland area lost) on a waterbody level.[12] After the calculation of CFs for all individual waterbodies, these values are aggregated on a watershed level, based on the catchment area of each individual wetland. The reason for this is that water consumption deprives all wetlands of water that are downstream of the point of consumption. Wetlands upstream of the point of consumption are, however, not affected. [12] For the surface water-fed wetlands we cover 80% of the land masses with characterization factors. [19] Impacts from water consumption cause all species to be lost on each m^2 of wetland area that is lost (contrary to land use). The original method of Verones et al. [11, 12]included a habitat rarity index, which was excluded here when harmonizing the methods (for land use, no such index exists).

In contrast to water use land use follows a top-down approach. The methodology applied for calculating the impacts on biodiversity from land use (Table 1) is based on de Baan et al.[14] and for compatibility reasons enhanced with a vulnerability score (*VS)* as described below in Equation 5 and the SI, S10. It applies a matrix-calibrated[20] species-area relationship (SAR) to calculate regional species loss for all global ecoregions. Based on this, CFs for land use were calculated per land use type *x* ('agriculture', 'pasture', 'managed forests', 'urban area', and 'natural habitat'), ecoregion and taxon. The land occupation impact quantifies the reduction of biodiversity during the land use phase, i.e. impacts that occur due to continuous suppression of biodiversity recovery while land is occupied by human activities. To be consistent with the water use assessment, marginal land use impacts were calculated, i.e. the additional species lost in an ecoregion if an additional square meter of land is used.

An overview of characteristics and assumptions is given in Table 1.

Table 1: Overview of modeling assumptions of land and water use assessment methodologies on a global scale.

	Water use	**Land use**
Reference	Verones et al.[11, 12]	de Baan et al.[14]
Unit of CF	Global species-eq·yr/m^3	Global species-eq/m^2·yr
Spatial Unit	Watersheds (upstream of wetlands): Average area 66 decimal degrees squared (standard deviation 117 decimal degrees squared)	Ecoregions: Average area 26 decimal degrees squared (standard deviation 197 decimal degrees squared)
Number of spatial units	233 (major watersheds, variation of CF within one watershed possible)	827 (ecoregions, no variation of CF within one ecoregion)
Addressing vulnerability of species?	yes	yes
Data for global species and VS maps	IUCN, BirdLife	WWF, IUCN, BirdLife
Modeling approach	marginal changes, non-linear classical species-area relationship Assumption that wetland area lost does not harbour any species any longer	marginal changes, non-linear matrix-calibrated species-area relationship accounting for the fact that converted ecosystems (matrix) may also provide some (typically lower) habitat quality for terrestrial species[20]
Type of impacts considered	immediate impact (analogous to occupation), no transformation included	immediate impacts (occupation), for transformation impacts see SI
Taxa covered	Amphibians, Birds, Mammals, Reptiles	Amphibians, Birds, Mammals, Reptiles
Current spatial coverage	Global (between 28% and 63 % of all wetlands, depending on global estimate of wetland area)[21]	Global (100% of terrestrial area)
Spatial resolution of CF	0.05°	Ecoregion

Effect factors (EF). The overall effect factor EF_t per taxon t and region a for both water use and different land use types (stressors j) is expressed as regional species loss (per ecoregion or wetland) per area of habitat lost, weighted by the vulnerability of the species present in that region. It is described with the following equation (Equation 3):

$$EF_{t,j,a} = \frac{\Delta S_{t,a}}{\Delta A_a} \cdot VS_{t,j,a} \cdot k_j$$

Equation 3

ΔS_t is the number of species lost per taxon t per unit of area lost in region a (ΔA_a), VS_t is the vulnerability score of all species of taxon t present in region a and k_j is an allocation factor.

The regional loss of species caused by all present land use types j is distributed to the individual land use types by using k_j. Thereby k_j depends on the land use types' habitat quality and area share.[14] Since for water use there is only one type, k_j equals one for water. ΔA_a is 1 m^2 for land use. For water use it is based on the calculation of wetland area changes caused bya consumption of 1 m^3/yr.[11] ΔS is calculated with actual data of species richness (on an ecoregion level for land use and on a 0.05° x 0.05° grid for water use) from WWF,[17] IUCN[22] and BirdLife[23] based on species-area relationships,[20, 24] as shown in Equation 4. $A_{old,a}$ refers to the natural extent of ecoregions (land use) or today's wetland area[11], $A_{new,a}$ is ecosystem area after the habitat conversion by land use or water consumption. The z-value (slope of the species-area relationship, see SI S3) is adapted in the matrix-calibrated SAR, in order to take into account that some species can survive in human-modified landscapes. Effect factors for each spatial unit a (ecoregion or wetland) were determined for four taxonomic groups t (birds, amphibians, mammals and reptiles).

$$\Delta S_{t,a} = 1 - \left(\frac{A_{new,a}}{A_{old,a}}\right)^{z_{t,a}}$$

Equation 4

The spatially explicit vulnerability score ($VS_{t,p}$) is calculated for all taxonomic groups and is used for both land and water use (Equation 5).[19] The *VS* is based on data from IUCN and BirdLife.[22, 23]

$$VS_{t,p} = \frac{\sum_{i=1}^{n} \frac{TL_{i,p} \cdot GR_{i,p}}{\sum_{i,p=1}^{m} GR_{i,p}}}{S_{t,p}}$$

Equation 5

This *VS* was calculated in Matlab[25] with a pixel resolution of 0.05° x 0.05°. The *VS*, as described in more detail in refs[12, 19], was calculated for each taxon t and each pixel p

individually, taking into account all species *i* within the taxon t with their distinct IUCN threat level (*TL*) and geographical range (*GR*, with a total of *m* pixels). The present species number *n* in each pixel is implicitly accounted for by summation, and to avoid double counting a division with the total, present species number of a taxon *S* is performed in each pixel. The *VS* accounts for differences in the vulnerability of species, with higher values for small-ranged (and thus intrinsically rare) species and for already occurring threats via the IUCN threat level.[26] For use in Equation 3, median *VS* values per ecoregion *a* were calculated for land use. Average values were calculated for the much smaller wetlands, because no extreme changes and outliers within the restricted wetland areas are expected. The *TL* were assumed to range on a linear scale from 0.2 (least concern) to 1 (critically endangered), excluding already extinct species. We calculated global CFs with and without *VS* for both water and land use and calculated correlation coefficients between them.

Aggregation factor (*W*). We propose four different options for aggregating impacts from different taxonomic groups and stressors.

<u>*Option 1:*</u> The first option gives equal weight to all species, no matter to what taxon they belong to, i.e. the impacts on individual species are simply summed (*W*=1). Owing to the different numbers of species per taxon (e.g. birds > 10'000, reptiles > 3000), some taxa will get a higher overall weight. In order to be able to compare these outcomes to the ones from the other options, a division with the overall, global species number of the four taxa (see SI, S6) is performed (Equation 6), resulting in a unit of globally potentially disappeared fraction of species (PDF_{global}):

$$W_t = \frac{1}{\sum_t S_{t,world}}$$

Equation 6

Where $S_{t,world}$ is the global number of species within the taxonomic group *t*. $S_{t,world}$ is equal to 10104 for birds, 5386 for mammals, 3384 for reptiles and 6251 for amphibians..

Option 2: The second approach gives equal weight to all taxonomic groups (i.e. it gives relatively less weight to species-rich taxa compared to option 1). It involves an aggregation according to the global species numbers $S_{t,world}$ of each taxonomic group *t* (Equation 7). The denominator here includes the number of taxonomic groups N, since the total $CF_{total,j,a}$ (equation 1) cannot be larger than 1 (=100% loss):

$$W_t = \frac{1}{N \cdot S_{t,world}}$$

Equation 7

$S_{t,world}$ is the absolute, global, actual species richness of taxonomic group *t*, for which spatial distribution information from the International Union for Conservation of Nature (IUCN)[22] or BirdLife[23] is available.

Option 3: The third approach weights taxonomic groups on the basis of the vulnerability of taxa (i.e. how threatened or rare species of a specific taxon are). A global vulnerability score ($VS_{t,world}$, Equation 8, concept based on Verones et al.[12]) is derived on a taxonomic group level, and combined with the species richness of each taxon (Equation 9). Note that $VS_{t,world}$ values are derived based on the $VS_{t,p}$ maps (from Equation 5) which are on a pixel level. The difference between $VS_{t,p}$ and $VS_{t,world}$ is that the former is calculated separately for each spatial unit (pixel, or aggregated to ecoregion), based on only those species present in this region, while the latter is summed over all global species of a taxa and their overall geographical range area.

$$VS_{t,world} = \frac{\sum_{p=1}^{m}\sum_{i=1}^{n}\frac{TL_{i,t} \cdot GR_{i,p,t}}{\sum_{p=1}^{m} GR_{i,p,t}}}{S_{t,world}} = \frac{\sum_{p=1}^{m} VS_{t,p} \cdot S_{t,p}}{S_{t,world}}$$

Equation 8

$VS_{t,world}$ is based on the IUCN threat level (TL) and the geographical range ($GR_{i,p,t}$, area in which the species can potentially be found, in km^2) of each species i of taxa t taking into account every pixel p where the species occurs (up to the total of m pixels). Implicitly, the number of species is included when summing all VS in a taxon for all species (up to n species), thus to exclude the richness, the VS is divided by the global species number $S_{t,world}$ of that taxon. The term $S_{t,world} \cdot VS_{t,world}$ in the denominator of equation 9 can be viewed as the threatened species richness of the taxa t. This option is also especially useful, if the overall impact shall be compared to impact categories that do not contain a vulnerability score themselves.

$$W_t = \frac{1}{N \cdot S_{t,world} \cdot VS_{t,world}}$$

Equation 9

Option 4: Finally, we propose a fourth approach that gives additional weight to the taxonomic groups with a higher vulnerability score $VS_{t,world}$ (equation 10), i.e. it weights taxonomic groups on the basis of the vulnerability of taxa (i.e. how threatened or rare species of a specific taxon are). This approach assumes that the loss of additional species from a taxa whose species are already rare and threatened (i.e. with a higher $VS_{t,world}$ value), results in higher global biodiversity damage.

$$W_{t,option\,4} = \frac{VS_{t,world}}{\sum_t S_{t,world} \cdot \sum_{t=1}^{4} VS_{t,world}}$$

Equation 10

Case study. To test the combined assessment and compare land and water use impacts on biodiversity, we applied the method to three important commodities (coffee, tea and sugarcane) produced across Kenya, a country rich in species diversity and threatened by land use and water consumption related to agricultural practice.[27, 28] We considered direct land and water use impacts of the crop cultivation stage, i.e. land occupation and water abstraction for irrigation. In 2010, coffee and tea were the most important export crops from Kenya according to their value (coffee: >204 Mio $, tea: >1 Billion $) and both were among the top five exported goods according to their exported weight (coffee: >43’000 t, tea: >415’000 t).[29] Sugarcane cultivation has been considerably increasing during the last decades,[30] and there are now about 60’000 ha of irrigated sugarcane area.[31] (In order to be able to assess both water consumption and land occupation simultaneously, we only take the area with irrigated sugarcane into account and exclude non-irrigated sugarcane). Sugarcane contributes about 15% to Kenya’s agricultural GDP.[30]

Coffee is mainly produced in areas where enough rain is falling for rain-fed agriculture, such as the highlands around Mt.Kenya, near Nakuru or in the Aberdare Range.[32] Tea is grown in higher elevations with sufficient rainfall, but also with irrigation, mostly around Mt. Kenya, the Aberdares, as well as the Kericho, Nandi and Kisii Highlands.[33] The area under sugarcane cultivation is mostly situated in the southern part of Kenya, such as in the Nyasa, Nyando and western sugar belt.[30] The distribution of the crop cultivation is shown in the SI (Figure S1). Sugarcane requires around 6340 m^3/ha of irrigation (based on refs[34, 35]). Tea and coffee are to a large extent rain-fed but may require some deficit irrigation. This was calculated for tea as 172 m^3/ha (based on refs[36, 37]) and for coffee as ca 310 m^3/ha (based on refs[34, 38]). Water consumption is assumed to stem from surface water only for all three crops. For the land use assessment, sugarcane is treated as an annual crop while coffee and tea are considered to be permanent crops in the relevant ecoregions (see SI). Yields are 0.29 t/ha for coffee, 2.2 t/ha for tea and 86.1 t/ha for sugarcane.[34] More details to land and water use inventories are presented in SI S4.

Results

Influence of vulnerability. To understand how much the results are affected by including vulnerability scores, we performed a correlation analysis using the global maps of CFs calculated with and without $VS_{t,p}$. The correlation varied between taxa and stressor. For surface water consumption, the lowest correlation was found for amphibians (0.27) and the highest for reptiles (0.92). Land occupation showed for the six land use types and all taxa a correlation between CF maps of 0.11 (birds, land use types: extensive forestry, pasture and urban areas) to 0.90 (mammals, land use type: intensive forestry). Of all 24 combinations of species and land use types for land occupation, 8 combinations had a correlation of more than 0.8 and 5 combinations a correlation of less than 0.2. A complete list of correlations is shown in SI Table S8.

Case study results for different aggregation options. Impacts from both land and water use were calculated for all three crops and for all four aggregation options using $VS_{t,p}$. The aggregation factors W_t for each taxon and option are displayed in Table 2.

Table 2: Aggregation scores W_t per taxonomic group for the three aggregation options (displayed as 10^{-5}).

	Option 1	Option 2	Option 3	Option 4
Reptiles	4.0	7.4	16.1	1.0
Birds	4.0	2.5	8.6	0.6
Mammals	4.0	4.6	10.4	1.0
Amphibians	4.0	4.0	6.8	1.3

The different aggregation options did not influence the ranking of crops according to their biodiversity impacts from water, land use, or both combined. Assessed per kg of harvested crop, coffee had the largest total impact (i.e. from land and water use) for all four aggregation options, closely followed by tea and sugarcane (see SI Table S8). For land occupation, coffee had the largest impacts for all options, while sugarcane had the largest water use impacts for all options (see SI, Tables S9 and S10). A comparison of the EFs for Kenya for land occupation and water use is shown in Figure 1 for aggregation option 2. Analogous figures for the other options are displayed in the SI (Figures S4, S5 and S6). The irrigated sugarcane cultivation lies within the area with the highest EF for water consumption. Sugarcane cultivation lies within a region with a slightly higher EF of land occupation, while tea and coffee are each in areas of somewhat lower impacts. However, since coffee has the largest area need per kg of crop, the impact of land occupation from coffee is still largest.

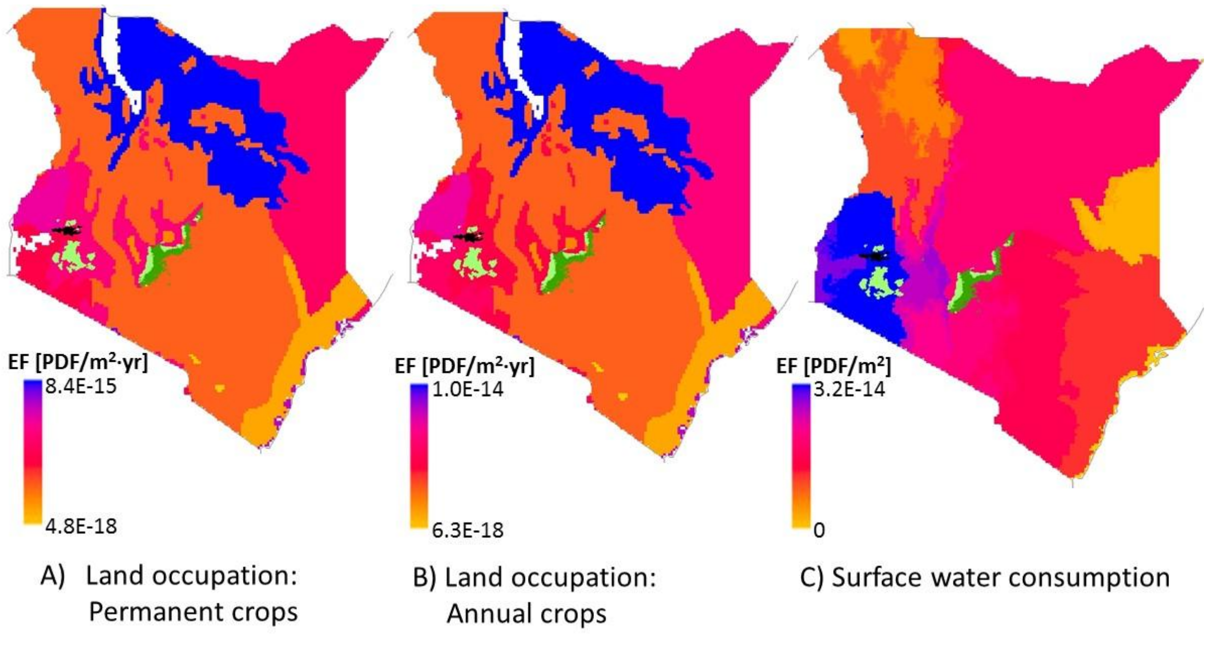
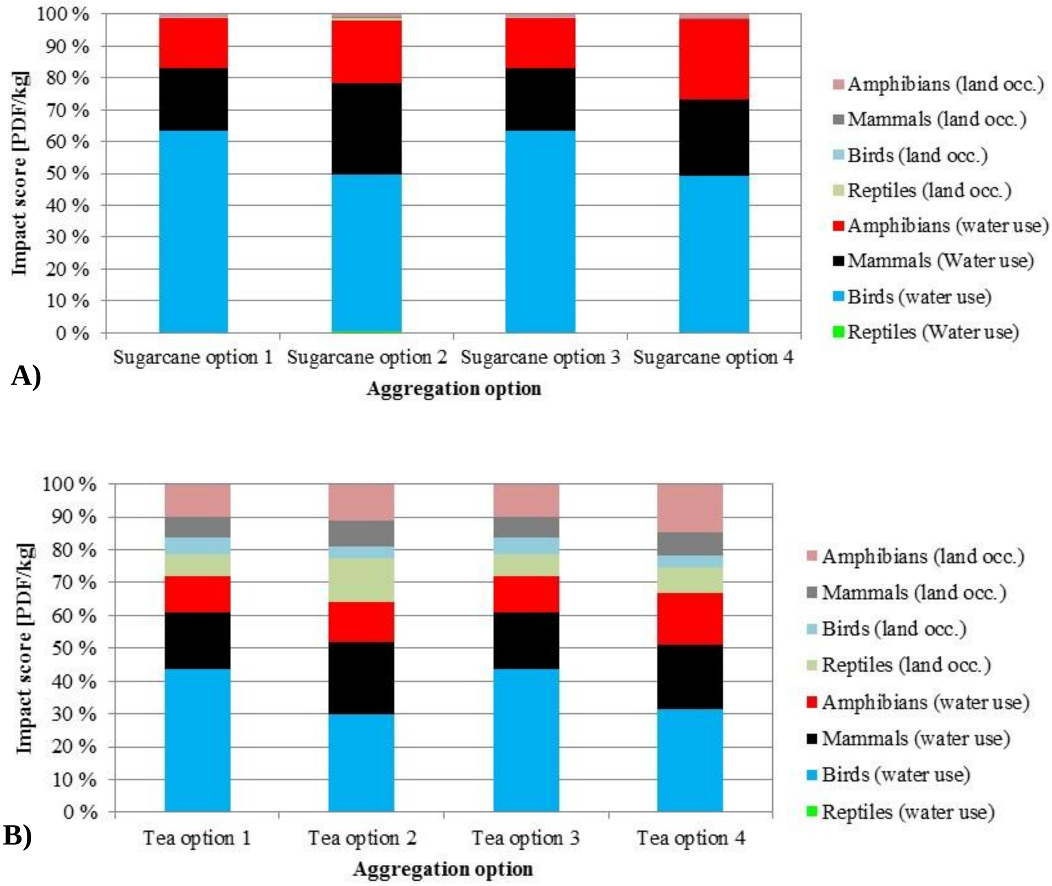


Figure 1: Spatially differentiated effect factors for A) land occupation from permanent crops, B) land occupation from annual crops and C) surface water consumption. Shown here are aggregated EFs over all taxa, aggregated with option 2. Outlined in dark green is the coffee area (permanent crops), in light green the tea area (permanent crop) and in black the irrigated sugarcane area (annual crop). Crop distribution is also shown in Figure S1 in the SI.

Impacts from water use dominated the overall impact score for sugarcane and tea, and land occupation was dominant for coffee (Figure 2). For all three options, sugarcane's impact was dominated by water use, irrespective of the consideration of the species vulnerability in the effect factor calculations (see SI and Figure 2). For tea, water use constituted between 63% and 72% of the impact (when neglecting land transformation), while for coffee land use dominated with at least 93% of the overall impact (see SI). For tea and coffee the importance of water use impacts increased to more than 97% and to more than 80%, respectively, if no $VS_{t,p}$ was included (see SI). For all aggregation options, birds, the most species rich taxon, dominated for sugarcane and tea. For coffee, reptiles (from land occupation) made the largest contribution to the impact score.

If land transformation impacts were considered, they contributed up to 57% of the overall impact for tea (see SI S9). For sugarcane and coffee it was not relevant, since these croplands did not expand or have hardly expanded in Kenya in recent years (SI, Tables S1 and S6).



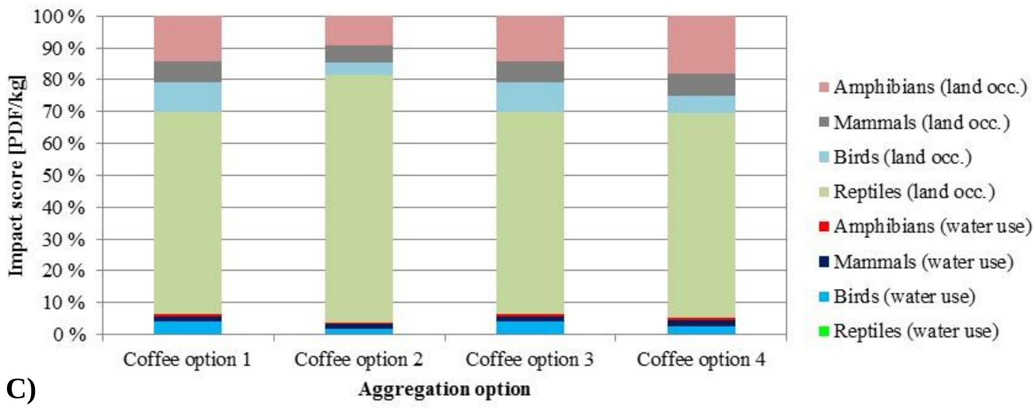


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Figure 2: The three graphs show the relative contribution to the impact score of the stressors water use and land occupation (land occ.) across the different taxa (amphibians, mammals, birds and reptiles). In each of the graphs, A) for sugarcane, B) for tea and C) for coffee, results for the four aggregation options are compared.

Discussion

Modeling procedure. As shown in Table 1, there are only slight differences between the methodologies applied for land and water use assessment, mostly related to the different spatial resolution of the CF.

An important difference between land occupation and water use impacts is the way in which species loss is accounted for. One m^2 of wetland area being dried up due to water consumption was assumed to result in the complete loss of species from that m^2, since all water is removed from that ecologically distinct area. However, if one m^2 of land is converted, it may not result in the total loss of species from that area, since the new land use may also harbor some species (see Koh et al.[20] for a detailed discussion). The adapted z-value accounts for the fact that not all species are lost per m^2 of land occupied, contrary to wetland area loss with standard z-values. In principle, the lost wetland area can provide habitat to terrestrial species, but this interaction was not considered here.

Other differences of the previous versions of the methods used here were eliminated to ensure consistency. For example, the effect factor (EF) of water consumption originally also considered habitat rarity, in addition to species vulnerability, which is now neglected.[12] For the land use assessment, land transformation was not considered in the main comparison for consistency. Transformation impacts provide information on the reversibility of an intervention, i.e. how fast an ecosystem recovers after land conversion or after reducing wetland areas due excess water abstraction. The habitat change is considered more damaging if the ecosystem only recovers slowly. Up to now, damage models for water consumption in LCA do not consider transformation impacts. While a water body might recover relatively quickly in terms of water volume if water consumption is stopped, this is unlikely for biodiversity. Such a recovery process of biodiversity has not been quantified for water use impacts so far, but this could be included analogous to land transformation (SI S9) once information about recovery times of wetlands becomes available. Assessing transformation impacts could be relevant in regions with large changes in land or water use, for example tea production in Kenya, which expanded considerably during the past 20 years. Here, land transformation impacts made up 57% of the total biodiversity impacts from both land and water. This highlights the necessity to further explore the possibility of incorporating transformation impacts also into water use impact assessments. Another issue that needs to be addressed in the future is the discrepancy between the spatial level of detail available for inventories and impact assessments. As mentioned in the beginning, irrigation can potentially decrease the amount of land used and thus create trade-offs.

The taxonomic groups we have considered are rather oriented towards terrestrial habitats, even though large numbers of species (especially amphibians) depend on water habitats as well. Other aquatic taxonomic groups such as fish have not been considered due to a lack of spatial data from IUCN. However, the principles of combining different taxa and stressors into a single biodiversity score can also be applied when data on more taxa become available in the future.

For our assessment, we chose ecologically relevant spatial scales for both land and water use, i.e. ecoregions and watersheds. In both cases, species loss in each region was modeled, but the size of the regions varied (Table 1, SI S7), and thus the regional species losses do not refer to the same size of "region". Ecoregions are more numerous than watersheds, and they vary more in size. These *regional* species losses were weighted by the vulnerability score of species, i.e. the threat for *global* extinction of species present per region, leading to the potentially lost fraction of global species. Therefore, we argue that the stressors can be compared despite the differences in spatial units.

Vulnerability. The relatively low correlations between CFs with and without VS highlight that in many cases the VS indeed provides additional information on biodiversity impacts from water and land use. The correlations were especially low for amphibians (for all stressors, i.e. correlation within water use, land occupation and land transformation). A reason for this large influence of the VS on that taxon is that amphibians in comparison to the other taxa have the smallest average geographical range sizes (see SI S6) and thus largest VS values. Also, the average, global VS of amphibians is highest, thus leading to the most pronounced changes in the CF maps. The correlation between CFs with and without VS was high for birds and in many cases also mammals. These species have on average larger geographical ranges and thus lower VS values (division with the geographical range area), decreasing the influence of the VS on the CF. The additional information gained through the VS is thus less pronounced in these cases.

Aggregation scenarios and case study results. There is no "correct" way of aggregating impacts from different stressor and taxonomic groups, as the different options contain different (normative) choices. Within the assessment of one crop, the ranking of the stressors does not change between the aggregation options, i.e. land occupation always dominates for coffee, and water use always dominates for tea and sugarcane (SI, Table S10). Also, the ranking between the crops in terms of overall impact does not change for the aggregation options (see SI Table S9).

The smaller impacts from land occupation on birds in the case of sugarcane are related to the fact that their EF is almost one order of magnitude smaller than for the other taxa for the considered production sites of sugarcane. However, it is also birds that are most affected by water use. In option 1 (all species have equal weight), this is because of the much higher species numbers of birds in comparison with the other taxa. In option 2 (all taxa have equal weight), the other taxa gain importance at the expense of the birds, highlighting that in this option there is no domination by the absolute number of species. In option 3 (including both species richness and vulnerability), birds gain some importance again, while the shares of very vulnerable taxa does not increase. However, in option 4 (strong focus on vulnerability)amphibians (taxon with highest $VS_{t,world}$) gain importance in both land occupation and water consumption.

Water use is the dominant impact for tea and irrigated sugarcane. The most important reason for this is that these crops are produced in areas with a large EF for water consumption, while the coffee (despite irrigation) is situated in areas with lower EFs. For land occupation, it is similar (see Figure 1, and Figures S4, S5 and S6), however due to the larger area requirement per kg of crop, the impact of coffee is still larger. The EFs for water use are often larger than for land occupation. This might be partly due to the fact, that for water use, all wetland species are lost within each m^2 of wetland area lost, while for land use some species survive on converted habitat. If $VS_{t,p}$ is not taken into account, the EFs of water are even larger. However, the maximum EF of land and water in Kenya are within a similar order of magnitude. The wetlands in Kenya and the surrounding watersheds are comparatively species-rich and thus of high ecological relevance, which explains the dominance of water use impacts in our case studies. Here, we only considered the irrigated sugarcane cultivating regions of Kenya. However, sugarcane is produced in other regions without irrigation as well. There, direct impacts of water consumption do not exist. To better understand the biodiversity impacts of crops produced in different regions of Kenya, better inventory data than used in this study would need to be collected (especially for yields, depending on the level of irrigation and land use intensity). However, for illustrating the approach for harmonization of biodiversity impacts across different stressors, the case study as described was considered sufficient.

The unit of the $CF_{total,j}$ is potentially disappeared fraction of species (PDF_{global}) per unit of intervention, while units of the previous versions of the methods were in absolute species loss. However, this PDF_{global} is not comparable with the PDF that has been until now commonly used in LCA. This new PDF_{global} is a global PDF, since it is based on the actual species richness and species vulnerability of all taxa involved. Thus, it is the potential fraction of species globally lost within the taxa considered, as opposed to previous PDFs that were based on local or regional loss with no attempt to quantify how many species were potentially lost globally and irreversibly. Furthermore, the result for all taxa and both stressors are based on the same principles, which makes the PDF_{global} compatible and thus helps to compare the biodiversity impact of land and water use.

Outlook. Each of the four aggregation options considers a different conservation concern and normative choice. Option 3 should be chosen in order to for example make impacts comparable with impact categories that do not take a *VS* into account. Option 1 should be chosen if equal weight should be placed on each species, disregarding the species richness of individual taxa. Species-rich taxa will dominate the impact score for this aggregation option and a comparison across stressors is only valid if data on the same set of taxonomic groups are available. Including additional taxonomic groups in only one stressor is only valid if we can exclude an impact on the taxonomic group from the other stressor (for instance fish could be included for impacts from water consumption, assuming that land use has no direct impact on fish). We therefore prefer option 2, which gives equal weight to each taxa instead of the single species and option 4, whose aim it is to target especially the taxa containing many vulnerable species, and at the same time to correct for the dominance of species rich taxa. However, the best aggregation option mainly depends on the goal of the study, and thus we leave it up to the user to choose the appropriate option for their application.

Since spatial differentiation and species information is becoming more readily available, it is also possible in the future to include other stressors in this framework, such as freshwater eutrophication or terrestrial acidification.[39, 40] This will contribute to a consistent aggregation and weighting of different impacts on ecosystems.

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